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Abstract

Feed in Tariffs (FiTs) and renewable heat incentives (RHIs) are driving a rapid expansion in anaerobic digestion (AD) coupled with combined heat and power (CHP) plants in the UK. Farm models were combined with consequential life cycle assessment (CLCA) to assess the net environmental balance of representative biogas, biofuel and biomass scenarios on a large arable farm, capturing crop rotation and digestate nutrient cycling effects. All bioenergy options led to avoided fossil resource depletion. Global warming potential (GWP) balances ranged from –1732 kg CO_2e Mg^{-1} dry matter (DM) for pig slurry AD feedstock after accounting for avoided slurry storage, to +2251 kg CO_2e Mg^{-1} DM for oil seed rape biodiesel feedstock after attributing indirect land use change (iLUC) to displaced food production. Maize monoculture for AD led to net GWP increases via iLUC, but optimised integration of maize into an arable rotation resulted in negligible food crop displacement and iLUC. However, even under best case assumptions such as full use of heat output from AD-CHP, crop-biogas achieved low GWP reductions per hectare compared with *Miscanthus* heating pellets under default estimates of iLUC. Ecosystem services assessment highlighted soil and water quality risks for maize cultivation. All bioenergy crop options led to net increases in eutrophication after displaced food production was accounted for. The environmental balance of AD is sensitive to design and management factors such as digestate storage and application techniques, which are not well regulated in the UK. Currently, FiT payments are not dependent on compliance with sustainability criteria. We conclude that CLCA and ecosystem services effects should be integrated into sustainability criteria for FiTs and RHIs, to direct public money towards resource efficient renewable energy options that achieve genuine climate protection without degrading soil, air or water quality.

Keywords: LCA; ecosystem services; anaerobic digestion; *Miscanthus*; GHG mitigation; land use change; renewable energy; biofuels

Introduction

Bioenergy trends and land use change

Heating, electricity generation and transport are major sources of greenhouse gas (GHG) emissions in industrialised countries such as the UK (Brown et al., 2012). Annually in the EU28, energy industries emit 1412 Tg CO_2e and the transport sector emits 926 Tg CO_2e (Eurostat, 2014). Bioenergy is anticipated to play a major role in meeting the European Union target for 20% of energy consumed to be from renewable sources by 2020, including 10% renewable transport fuels (EC, 2009). Mandatory biofuel blend targets and incentive schemes such as duty exemption for biofuels, electricity feed-in-tariffs (FiTs), capital grants and renewable heat incentives (RHIs) are being implemented to encourage bioenergy throughout the world (HPLE, 2013). Global biofuel production in 2011 amounted to 100 billion litres, largely from food crop feedstocks, giving rise to concerns over food price increases and land use change pressures (HPLE, 2013). Policy and commercial development is now shifting to "second generation" biofuels produced from lignocellulosic feedstocks that may alleviate competition with food production. However, currently in the UK there is concern that financial incentives for anaerobic digestion (AD), including FiTs of up to €0.188 per kWh for biogas electricity (FIT Ltd, 2013) and the new RHI (Ofgem, 2013), could lead to the appropriation of large areas of arable land to grow crop feedstocks such as maize (Mark, 2013). In Germany, over 1,157,000 ha of land are used to grow crops for AD (FNR, 2013).

Almost 60% of land required to produce products consumed within the EU is located outside of the EU (Tukker et al., 2013), and global demand for agricultural commodities is rising rapidly (FAO Stat, 2014), so there is little "spare" land available for bioenergy feedstock cultivation (Dauber et al., 2012). Feedstock production for bioenergy is driving land use change (LUC) at a global level (HPLE, 2013; Warner et al., 2013). Indirect land use change (iLUC) associated with the displacement of food production by bioenergy crops may cancel or exceed GHG emission mitigation achieved via fossil energy substitution (Tonini et al., 2012; Hamelin et al., 2014). It is therefore important that possible iLUC effects are accounted for in sustainability assessment of bioenergy options.

Consequential life cycle assessment

Attributional life cycle assessment (ALCA) is an increasingly popular systems approach used to quantify resource flows and environmental burdens arising over the value chain of a product or service (ISO, 2006a; b). Environmental impact categories relevant to agricultural systems include global warming potential (GWP), eutrophication potential (EP), acidification potential (AP) and fossil resource depletion potential (FRDP). The EU Renewable Energy Directive (RED) (EC, 2009) bases GWP sustainability thresholds for biofuels on ALCA calculations.

Accounting for global net effects of bioenergy production arising from factors such as iLUC and diversion of organic waste streams requires a consequential LCA (CLCA) approach. CLCA expands system boundaries to account for marginal effects of system modifications induced via economic signals throughout the wider economy (Weidema, 2001). CLCA is increasingly being applied to assess bioenergy (e.g. Mathiesen et al., 2009; Dandres et al., 2011; DeVries et al., 2012; Hamelin et al., 2012; Rehl et al., 2012; Tonini et al., 2012; Tufvesson et al., 2013; Hamelin et al., 2014; Styles et al., 2014). Displaced food production can be complicated to model within CLCA because it gives rise to a mix of intensification, land transformation and cascading displacement of crops (Schmidt, 2008; Kløverpris et al., 2008; Mulligan et al., 2010). These consequences can be estimated from market data or general equilibrium economic models, with high uncertainty (Schmidt, 2008; Earles et al., 2012; Marvuglia et al., 2013). Zamagni et al. (2012) argue that CLCA can lead to opaque and misleading outputs. However, the use of simplified, qualitative scenarios (Schmidt, 2008; Marvuglia et al., 2013; Vazquez-Rowe et al., 2014), can improve the transparency and insight provided by CLCA, if uncertainty is acknowledged. Accordingly, this paper presents results for a range of simplified best- to worst- case scenarios that span the range of plausible bioenergy situations for UK arable farms.

Farm modelling

Globally, agriculture and related LUC is responsible for 30% of global anthropogenic greenhouse gas (GHG) emissions (IPCC, 2007a). Agriculture accounts for 94% of ammonia (NH_3) emissions in Europe (EEA, 2012), the majority of diffuse nutrient losses to water (EEA, 2010), and relies on finite resources of phosphate for fertilization (Cordell et al., 2009). Farm scale AD can reduce GHG emissions from manure management and organic waste disposal whilst displacing fossil energy carriers, and associated GHG emissions, with the renewable biogas produced. Digestate from AD plants is also a useful fertiliser, but can lead to elevated NH_3 emissions during storage and spreading (Rehl & Müller, 2011). Importing municipal and commercial organic wastes into farm scale AD can considerably improve economic viability and increases GHG mitigation via the avoidance of landfilling and composting (Mistry et al., 2011a; Styles et al., 2014). Anaerobic digestion fundamentally alters resource flows on farms, with important implications for nutrient cycling and GHG emissions, whilst the introduction of new crops can lead to changes in crop rotations and soil C equilibria. Thus, in addition to boundary expansion via CLCA, accurate accounting for the net environmental effects of bioenergy production requires farm-system modelling that goes beyond default IPCC emission factors or standard unit process data available in commercial LCA databases (Del Prado et al., 2013). There remains a need to assess how AD could affect nutrient cycling, land use and crop rotations on typical arable farms.

Recently, Styles et al. (2014) described a novel combination of farm modelling, CLCA and bioenergy scenarios embodied within the “LCAD” tool (Defra, 2014). Using CLCA to capture net changes for plausible but simplified farm bioenergy scenarios provided transparent insight into the risks and opportunities associated with particular AD feedstock and management options on dairy farms. In this paper, we employ the same method to evaluate bioenergy scenarios for arable farms.

Ecosystem services assessment

Ecosystem services (ES) are defined as the outputs of ecosystems from which people derive benefits, considered under the broad headings of provisioning, supporting, regulating and cultural services (Mace et al. 2011). Enclosed farmland is managed primarily for the provisioning of food but is important for many other ES which can be heavily impacted by changes in cropping pattern (Firbank et al. 2013) and management practices (Zhang et al., 2007; Power, 2010). Such effects depend on landscape context, and are not well represented in traditional LCA – although LCA methodologies are being developed to account for important ecosystem factors such as soil quality and water flow/quality regulation (Cowell et al., 2000; Maes et al., 2009; Zhang et al., 2009; 2010; Saad et al., 2011; Oberholzer et al., 2012; Garrigues et al., 2013). The UK National Ecosystem Assessment (Mace et al., 2011) provided a framework for the classification and assessment of ES that may be applied alongside LCA in a qualitative manner to highlight major environmental effects not detected by traditional LCA methodology.

Aims and objectives

In this paper, we summarise the outputs from farm models coupled with CLCA, supplemented with a screening of major ES effects, to comprehensively compare the environmental sustainability of biogas, biofuel and biomass options on arable farms. Multiple data sets were integrated within the "LCAD" scenario tool developed to inform policy makers and prospective farm AD operators on the net global environmental effects of plausible farm bioenergy scenarios (Defra, 2014).

The objectives of this study are to: (i) quantify the net environmental effects of plausible bioenergy scenarios and feedstocks on arable farms; (ii) assess the influence of AD design and management factors on environmental performance; (iii) compare the land- and economic- efficiency of GHG mitigation via different bioenergy pathways; (v) highlight bioenergy ecosystem services effects not reflected in LCA metrics.

Materials and methods

Scope and boundaries

This study presents CLCA and ALCA results generated by the LCAD tool that underwent review by expert members of a technical working group (TWG, 2013), and is available online (Defra, 2014). A modified iLUC module was added to the tool for this study. The primary CLCA outputs are calculated as net change in annual environmental burdens calculated after accounting for major processes directly and indirectly influenced by the introduction of bioenergy options into a baseline arable farm system. The cultivation of crops for food and animal feed production ("food crops") is held constant, but displaced elsewhere where bioenergy crops are cultivated, so that one year of food crop production on the baseline farm is the primary functional unit. As per CLCA methodology, all displaced and replaced processes are accounted for as additional environmental burdens (debits) or avoided environmental burdens (credits) (Figure 1). In addition to displaced food crop production (debit), processes replaced (credits) in bioenergy scenarios include: (i) marginal UK grid-electricity generation via natural gas combined cycle turbines (NGCCT) (DECC, 2012); (ii) heat generation via oil boilers; (iii) petrol and diesel combustion; (iv) composting of food waste; (v) high-protein animal feed production; (vi) fertiliser manufacture and application. Environmental burdens for important upstream and counterfactual processes are detailed in Table 1. [Insert Figure 1 and Table 1 about here]

Infrastructure is excluded from the scope, as per EC (2009) and BSI (2011) for GHG accounting. The temporal scope is approximately 10 years, considering the time required for wider adoption of farm bioenergy options and current prevailing technologies for counterfactual processes. The geographic scope is global. Four environmental impact categories are accounted for based on CML (2010) characterisation methodology (Table S1.1). We present results for a range of simplified narratives generated as scenario permutations within the LCAD tool (Table 2). Default results are based on the typical UK situation (TWG, 2013), but results are also expressed as a full range of possible outcomes representing worst- to best-case scenario permutations (Insert Table 2 about here).

Environmental effects are calculated as the net difference (global change) between annual environmental burdens calculated for the baseline farm and for the bioenergy scenarios, expressed as annual pollutant loadings and percentage change. Environmental burden changes are also calculated per Mg dry matter (DM) of bioenergy feedstock produced, per hectare farm area appropriated for bioenergy crop cultivation, per MJ lower heating value (LHV) of feedstock and per MJ useful energy output. For comparison with CLCA values and GWP sustainability thresholds set out in the RED (EC, 2009), ALCA burdens are calculated per MJ fuel energy output based on process separation within the farm model and energy allocation.

Farm models

The baseline farm (A-BL) is defined as a large (400 ha) arable farm in the East of England, based on a typical four year rotation (FBS, 2013): 100 ha each of first winter wheat, second winter wheat, spring barley and oil seed rape (OSR) (see Data S2.1). The baseline farm was parameterised according to economic optimisation within the Farm-adapt model (Gibbons et al., 2006) based on recommended fertiliser (NPK) application rates for UK crops (Defra, 2010) and average yields for good quality arable soils (Nix, 2009). A derivative of the standard baseline farm (AP-BL) is used for a pig-slurry plus food waste AD scenario (AD-SF) (see Data S2.2). For both AP-BL and AD-SF it is assumed that 5098 Mg of pig slurry is transported 8 km in a tractor tanker from a typical intensive pig farm (Newell-Price et al., 2012). Pig slurry is applied to the first winter wheat rotation in September at a rate of 22 Mg/ha and to the spring barley rotation in April at a rate of 30 Mg/ha, replacing fertiliser according to nutrient availability after leaching and volatilisation losses calculated in the MANNER NPK tool (Nicholson et al., 2013).

Mineral fertiliser application rates for baseline farms and scenario farms were calculated from crop nutrient requirements (Defra, 2010) minus plant-available nutrients delivered by pig slurry and digestate applications determined by MANNER-NPK (Nicholson et al., 2013) – elaborated in Data S2. Diesel consumption for field operations was calculated in Farm-adapt based on hours of field operation. The embodied burdens attributed to major inputs to the farm, and key counterfactual processes were taken from Ecoinvent (2010) and other sources (Table 1).

Direct emission factors are summarised in Table 3. Field losses of NH_3 and NO_3^- from slurry and digestate applications were calculated in MANNER-NPK, assuming a broadcast application of pig slurry and shallow injection application of liquid digestate. Direct and indirect N_2O-N emissions were calculated as per IPCC (2006). For tractor diesel combustion, NO_x emissions were approximated to EURO III emission standards for 75-130 kW off-road vehicles assuming 30% engine efficiency (Dieselnet, 2013). [Insert Table 3 about here].

Counterfactuals and iLUC

Table 1 summarises environmental burdens for the major counterfactual products and processes considered in this study. Here we elaborate some important counterfactual assumptions. In-vessel composting and landfill are the main fates of food waste in the UK (Mistry et al., 2011a), for which environmental burdens were modelled in Styles et al. (2014). Food waste going to landfill is declining rapidly in response to economic and regulatory drivers being implemented under the Waste Framework Directive (2008/98/EC), and farm AD requires separated organic waste fractions, which are less likely to go to landfill than unsorted municipal waste. Therefore, composting is the default counterfactual option for food waste, but landfill with 70% biogas capture and electricity generation was modelled as an alternative counterfactual to generate best case AD scenarios.

Bioethanol and biodiesel production from wheat and OSR result in high-protein dried distillers grains with solubles (DDGS) and rape seed cake (RSC) co-products. These co-products were assumed to replace a mix of soybean meal (marginal protein feed) and maize silage (marginal energy feed) calculated to deliver the same quantities of crude protein and metabolisable energy according to a feed ration calculator (EBLEX, 2014). Soybean meal substitution incurs knock-on displacement effects via soy oil substitution of palm oil, with implications for net iLUC. Details are given in DataS3.2.

Direct and indirect LUC GHG emissions and N mineralisation were calculated according to IPCC (2006) tier 1 methods (Data S3.2). The maximum possible (worst case) areas of global iLUC incurred for each bioenergy scenario were calculated as the area of food crop production displaced on the arable farm, minus the net area avoided from animal feed substitution by biofuel co-products. All iLUC was assumed to occur at the global agricultural frontier, which was defined as native grassland in Argentina and forest in Brazil, Indonesia, Thailand and Angola according to the five countries showing the greatest expansion in agricultural area over the past five years (FAO Stat, 2014). The iLUC method is elaborated in Data S3.2. An alternative iLUC method is proposed in Data S3.3, and provides the basis for sensitivity analysis.

<u>Bioenergy scenarios</u>

Eight plausible bioenergy scenarios were developed, reflecting recent reports (Mistry et al., 2011a; b; Defra, 2011), a farm AD visit and expert feedback (TWG, 2013). Two typical transport biofuel chains and one possible biomass heating chain were modelled to compare the relative efficiency of AD options (Table 4). Farm-adapt was used to optimise the integration of the bioenergy feedstock into the rotation (Figure 1; Table 4; Figures S4.1 to S4.7). Additional agronomic information is contained in Data S2.5.

[Insert Table 4 about here]

Key points are summarised below.

- AD-F: A quantity of 10 000 Mg food waste is imported to an on-farm AD unit, constrained by K_2O surplus (the first nutrient to reach surplus in available form) (Figure S4.1).
- AD-MZ_{rot}: 10% of farm area (40 ha) is used to cultivate maize, integrated into an optimised rotation where maize acts as a break crop, enabling 40 ha of lower-yielding spring barley (Table S1.2) to be replaced with 40 ha of higher-yielding first winter wheat, with a reduced yield because of delayed sowing, so that farm food production is reduced by just 1% (Figure 1).

Maize is supplied to an AD unit supplied by multiple farms that fuels a 1MWe combined heat and power (CHP) generator. This represents a best case scenario for maize-only AD.

- AD-MZ_{mono}: 100% of farm area is used to grow maize continuously in monoculture to feed an on-farm AD unit. This represents a more typical maize-only AD scenario, based on large areas dedicated to AD-maize cultivation in Germany (FNR, 2013) (Figure S4.2).
- AD-G: 10% of farm area (40 ha) is used to cultivate rye grass, displacing 10 ha of each crop in the four year baseline rotation to supply a multi-farm 1 MWe AD-CHP system (Figure S4.3).
- AD-SF: 5098 Mg of pig slurry is co-digested with 6000 Mg of food waste in an on-farm digester, constrained by nutrient demand for K_2O (Figure S2.4). Avoided slurry storage emissions from the pig farm are accounted for as an AD credit (see Data S1.2 and Figure S4.4).
- H-M: 10% of farm area (40 ha) is used to cultivate *Miscanthus*, transported 50 km to a pelleting factory, then a further 50 km to combustion in commercial biomass boilers, replacing oil heating (Figure S4.5).
- Eth-WW: 100ha of first winter wheat is used as a feedstock for bioethanol. DDGS co-produced alongside ethanol replaces soybean meal and maize on an equivalent protein and energy content basis (Figure S4.6 and Data S3.2).
- Bio-OSR: 100 ha of OSR is used as a feedstock for biodiesel. RSC co-produced with biodiesel replaces soybean meal and maize on an equivalent protein and energy content basis (Figure S4.7 and Data S3.2).

<u>Bioenergy conversion</u>

Five AD design and management options were modelled to reflect the important influence of fermentation efficiency and fugitive emissions from fermenters and digestate storage tanks on environmental performance (Table S4.1). Central results in this study are based on default parameters in Table S4.1, with best- and worst- case parameters used to generated performance ranges. NH_3-N emissions are calculated as a fraction of total ammonical nitrogen (TAN) present in the digestate, up to 10% in the case of open-tank storage (Misselbrook et al., 2012). We assume 5% of the CH_4 yield is emitted to the atmosphere during open-tank digestate storage (Jungbluth et al., 2007), and 2.5% of the CH_4 yield is emitted to the atmosphere during closed tank storage (TWG, 2013). The characteristics of the four feedstocks and associated post-AD digestate, which have important implications for fugitive emissions and fertiliser replacement, are summarised in Data S2.4. Arable farms typically have low heat demand, so under default LCAD settings heat output from the CHP is used to heat the AD process and for pasteurisation of digestate containing food waste where relevant, and the remainder is dumped. This is typical of AD-CHP units in the UK (TWG, 2013).

Miscanthus pellets replace oil heating, after *Miscanthus* biomass is transported 50 km from the farm to the pelleting plant, and pellets transported a further 50 km to the final consumer. Pellet processing consumes 240 kWh electricity, and 300 kWh of oil heating, per Mg DM (Anonymous, 2013). One Mg DM *Miscanthus* contains 18 GJ LHV, and displaces 16.2 GJ LHV of delivered oil-heat. Pellet boiler combustion emissions of NO_x and SO_x were calculated based on thresholds reported by the Biomass Energy Centre (2013): 120 mg NOx per MJ and 20 mg SOx per MJ.

Following calculation of feedstock cultivation burdens in the farm model, burdens for processing and transport of biofuels were calculated by multiplying activity data from Biograce (2012), assuming natural gas and electricity energy carriers, by Ecoinvent (2010) process burdens. Biofuels replace petrol and diesel on an energy basis. Direct combustion emissions of NO_x were assumed to be the same for fossil- and bio-fuels.

Economic and ecosystem services assessment

GHG abatement costs were calculated for each scenario, based on net margin changes on the bioenergy farm, plus net margin changes for the biofuel wholesaler and biomass end user, divided by the lifecycle GHG abatement achieved for each scenario. These theoretical marginal abatement costs equate to the support value required for bioenergy chains to break even with counterfactual food crop, energy generation and waste management systems. Economic assessment is elaborated in S5. An ES screening exercise was undertaken to describe effects not well captured by the LCA methodology applied (Data S6).

Results

Bioenergy scenario results

The magnitude of change relative to the baseline farm depends on the scenario-specific quantity of bioenergy generated, in addition to the environmental efficiency of each bioenergy option (Figure 2 and Table 5). Excluding iLUC, all scenarios result in a net GWP reduction compared with the counterfactual baseline. However, the GWP balance for maize monoculture (AD-MZ_{mono}), grass AD (AD-G), bioethanol (Eth-WW) and biodiesel (Bio-OSR) is positive (i.e. results in a net GHG emission increase) under the default assumption that 50% of displaced food production incurs iLUC. Eutrophication and acidification burdens increase across all scenarios that involve cultivation of bioenergy crops, but decrease substantially in the food waste and pig slurry scenarios owing to avoided waste and slurry management (Table 5). The magnitude of avoided resource depletion is proportionate to fossil energy substitution, and, for AD-MZ_{mono} under absolute best case assumptions, equates to 11 times the resource depletion on the baseline farm. [Insert Figure 2 and Table 5 about here].

Results for GWP and acidification are sensitive to whether or not CHP-heat is wasted or used to replace oil heating, and to AD design and management parameters that influence fugitive emissions of CH_4 and NH_3 (Figure 2). The reduction in acidification burden associated with digestion of waste (food waste and slurry) feedstock varies by a factor of four, according to management practice, reflecting the high NH_4-N content of relevant digestates. However, the GWP burden changes for maize monoculture and grass AD remain positive (i.e. GHG emissions increase) even under best case AD design and management with use of all CHP-heat under the default assumption that 50% of displaced food production incurs iLUC (Figure 2).

The environmental balance of waste digestation is highly sensitive to the type of waste management avoided. With a capped landfill rather than a composting counterfactual, the GWP reduction in the AD-F scenario increases by two-fold, reflecting avoided landfill CH_4 leakage, but acidification and eutrophication burdens increase, reflecting higher NH_3 emissions from digestate storage and land spreading than from landfilling.

Environmental efficiency of bioenergy feedstocks

The environmental balance of different bioenergy feedstock options on a Mg DM basis is compared in Figure 3. Fossil energy substitution makes a modest contribution to GWP burden changes, but makes only minor contributions to eutrophication and acidification burden changes. Credits arising from reduced on-farm food production are cancelled by debits arising from displaced food crop cultivation, and the iLUC debit associated with the latter makes a substantial contribution to the GWP balance of all crop feedstocks except for maize-in-rotation (Figure 3 and Tables S7.1 to S7.4.) Accounting for 50% iLUC, the GWP balance per Mg DM feedstock ranges from –1732 kg CO_2e for pig slurry to +2251 kg CO_2e for oilseed rape used for biodiesel production (Figure 3a). Notable GWP, acidification and eutrophication credits are attributable to the avoidance of food waste composting and pig slurry storage. Grass and *Miscanthus* lead to significant on-farm soil C sequestration (direct LUC) GWP credits that somewhat offset iLUC GWP debits. [Insert Figure 3 about here].

Feedstock cultivation and displaced food production dominate eutrophication burdens in most scenarios. Avoided animal feed production leads to significant GWP and eutrophication credits per Mg grain and oil seed used for biofuel production. These credits include avoided iLUC but do not fully offset the GWP debits incurred by displaced wheat and OSR production. Fugitive emissions of NH_3 from digestate storage and field application significantly influence eutrophication and acidification burden changes for food waste and pig slurry in the AD-F and AD-SF scenarios (Table S7.2 and S7.3). Imported nutrients applied in digestate lead to lower fertiliser manufacturing burdens for the AD-F and AD-SF scenarios, but higher soil emissions in the AD-F scenario (Tables S7.1 to S7.4). The acidification burden of food production declines following digestion of slurry owing to the assumption that field application technique changes from splash-plate for counterfactual slurry application on the AP-BL farm to injection application of digestate in the bioenergy scenario.

Cropping area GHG mitigation efficiency

Excluding iLUC effects, crop AD achieves GHG mitigation of 1.3 to 3.5 Mg CO_2e yr^{-1} per hectare of land planted with maize or grass, more than the small mitigation achieved by wheat bioethanol and oil seed rape biodiesel, but considerably less than the 21.5 Mg CO_2e yr^{-1} mitigation per hectare of *Miscanthus* grown to produce heating pellets (Figure 4). Only maize in rotation and *Miscanthus* achieve net GHG mitigation when iLUC is attributed to 50% of displaced food production, of 1.4 and 9.1 Mg CO_2e ha^{-1} yr^{-1}, respectively. Monoculture maize and grass AD and the biofuel options lead to substantial GHG emission increases of between 3.15 and 11.44 Mg CO_2e ha^{-1} yr^{-1} when 50% iLUC is accounted for (Figure 4). Bioethanol and biodiesel are less sensitive to iLUC than the other options because the animal feed substitution credits increase with the iLUC ratio. This effect is proportionately greater in the alternative iLUC method (Method 2), in which soybean and palm oil iLUC factors were higher than displaced wheat iLUC factors (S3.3). The method of iLUC estimation only affects the ranking of (less-bad) bioenergy options in terms of GHG mitigation per hectare under 100% iLUC, when *Miscanthus* leads to a net GHG emission increase according to the default method 1 but not according to alternative method 2.

The percentage of displaced food production that would need to incur iLUC in order to cancel any GHG abatement is: 5% for maize in the AD-MZ_{mono} scenario, 14% for grass in the AD-G scenario, 85% for *Miscanthus* in the H-M scenario, 5% for wheat in the Eth-WW scenario and 2% for OSR in the Bio-OSR scenario. [Insert Fig. 4 about here].

GHG mitigation costs

The AD-F and H-M scenarios result in net margin increases before subsidies, and all other default scenarios except AD-G are profitable after application of FiT and RHI subsidies (data not shown). Net post-subsidy losses for farmers who grow *Miscanthus* are outweighed by savings for end-users compared with oil heating. Minimum theoretical CO_2 abatement costs, based on subsidy needed for bioenergy chains to break even, vary from -€38 Mg^{-1} CO_2 for *Miscanthus* heating to €1189 Mg^{-1} CO_2

for AD-MZ_{mono}, under default settings excluding iLUC and use of CHP heat (Table 6). GHG mitigation costs for the AD scenarios reduce significantly if all net CHP heat output replaces oil heating, but AD based on slurry/food waste and *Miscanthus* heating pellets maintain a significant advantage over the AD-MZ_{rot} scenario and a large advantage over other bioenergy crop options.

Attributional versus consequential LCA results

GWP burdens per MJ biofuel produced are presented in Table 6, based on CLCA and also ALCA methodology for comparison with Renewable Energy Directive threshold values (EC, 2009). Accounting for possible iLUC effects within CLCA increases the GWP burden of biofuel production by a factor of between 3 and 8 for the AD-MZ_{mono}, AD-G, Eth-WW and Bio-OSR scenarios (Table 6). The CLCA approach also leads to negative CO_2e values per MJ biogas produced from food waste and pig slurry, reflecting credits associated with counterfactual waste management and slurry storage that outweigh the transport and fugitive CH_4 emission debits. The former credits are not accounted for in ALCA methodology. The CLCA approach also captures the displacement of animal feed by biofuel co-products, an effect that actually leads to a higher biofuel GWP burdens compared with ALCA based on allocation because avoided SBME production leads to avoided soy oil production which leads to more GHG-intensive palm oil production (Data S3.2).

Ecosystem services effects

The ecosystem services effects for each of the scenarios requiring land for bioenergy crop production are summarised in Table 7 and described fully with supporting references in Data S7.2. Maize scenarios are associated with strong negative effects owing to soil compaction, erosion, humus depletion, water runoff and low biodiversity. However, where maize extends very short crop rotations, some positive effects on habitat function and species richness could arise at the landscape level. Amongst the bioenergy crops, *Miscanthus* has the most positive portfolio of effects (Table 7), potentially leading to soil and water quality benefits, and biodiversity benefits when managed extensively. However, there is a risk that any positive local effects for the bioenergy crop scenarios identified using ecosystem services assessment may be offset by indirect effects associated with displaced food production, especially iLUC, that are not captured in the ecosystem services assessment methodology.

Discussion

Environmental balance of farm bioenergy options

Consequential life cycle assessment of farm bioenergy scenarios confirmed that biogas production from farm and food wastes and Miscanthus heating pellet production can achieve significant GHG mitigation and fossil energy substitution, but can give rise to additional eutrophication and acidification burdens. In the case of anaerobic digestion, acidification burdens can be minimized by well-sealed digestate storage tanks and injection application of digestate. In the longer term, the benefits of on-farm food waste digestion are likely to decline as prevailing waste management options move towards more efficient techniques such as mechanical and biological treatment coupled with anaerobic digestion (Montejo et al., 2013) or integrated waste refineries (Tonini et al., 2013).

Crop-biogas, bioethanol from wheat and biodiesel from oil seed rape can contribute to energy security at the expense of food security, but are neither land- nor cost- efficient options for GHG abatement compared with miscanthus heating pellets and waste-biogas, and risk significant increases in global GHG emissions through indirect land use change. Crop-biogas and liquid biofuel options are also associated with possible ecosystem dis-services at the landscape scale, especially soil degradation and associated reductions in water quality and availability in the case of maize. However, introducing limited areas (c.10%) of bioenergy cropping into short food-crop rotations could in some cases present an opportunity to improve rotation efficiency, somewhat mitigating the risk of indirect land use change.

Environmental assessment of on-farm bioenergy options

This study highlights the importance of considering food production and waste management displacement effects via consequential LCA when assessing the environmental balance of bioenergy options, building on similar conclusions from recent studies (e.g. Rehl et al., 2012; Tonini et al., 2012; Tufvesson et al., 2013). These effects fundamentally alter conclusions about the environmental balance of different bioenergy options, especially for global warming and eutrophication burdens. In addition, this study demonstrates the value of using farm models to identify opportunities for optimised integration of bioenergy feedstock cultivation within crop rotations, and to capture pertinent nutrient cycling effects associated with digestate use that are often omitted in attributional LCA and simplified in consequential LCA (e.g. Boulamante et al., 2013). The environmental effects of animal feed co-production with transport biofuels are also more accurately represented in consequential LCA than via allocation in attributional LCA. This study counters the findings of Weightman et al. (2011), who attributed a large GHG credit to bioethanol production, reflecting land use change avoided through DDGS substitution of soybean meal, but did not account for indirect land use change attributable to the displacement of food-wheat production.

The CLCA framework highlights that bioenergy crop cultivation always leads to higher eutrophication burdens, because more fertiliser must be applied globally to maintain food and bioenergy crop production. This important trade-off with GHG and resource depletion benefits is often overlooked in attributional LCA studies which consider only (often relatively low) direct fertiliser application to bioenergy crops (e.g. Styles & Jones, 2007). The coupled farm-model and consequential LCA approach greatly facilitates more complete and accurate framing of complex displacement issues via simplified transparent narratives that avoid uncertain and sometimes opaque macro-economic modelling associated with regional scale consequential LCA (Schmidt, 2008; Zamagni et al., 2012). These narratives provide insight into the pathways that link particular bioenergy policy or management decisions with environmental risks and opportunities.

Changes in cropping patterns arising from bioenergy feedstock cultivation can lead to significant ecosystem service effects not well captured within LCA, including soil erosion risk, water provisioning and flood regulation effects. These effects appear to be important for some bioenergy feedstocks such as maize, and therefore should be screened for during bioenergy sustainability assessment.

Sustainable bioenergy policy

Subsidies such as FiTs and RHIs, and mandatory biofuel blend targets, underpin the financial viability of all the bioenergy options considered here. FiTs provide essential support for the deployment and development of renewable energy options in energy markets still dominated by polluting fossil fuels. However, FiT payment is not dependent on the sustainability of bioenergy feedstock or transformation options (FIT Ltd, 2013), which has led to a high share of crop feedstock and a low rate of heat utilization for new biogas-CHP units in the UK (NNFCC, 2014), with poor environmental outcomes. Especially where crop feedstock is required, the use of public money should be tied to robust sustainability criteria based on consequential LCA and ecosystem service assessments in order to deliver maximum public benefit. Such assessment should consider how bioenergy crops fit into crop rotations in order to determine the magnitude of possible food displacement and indirect land use change.

The design and management of biogas plants also requires policy steer to avoid possible negative environmental outcomes. For ammonium-rich digestates derived from food waste and slurry feedstocks in particular, covered storage and injection application of digestate should be encouraged or mandated to minimise eutrophication and acidification burdens caused by ammonia emissions.

Miscanthus has considerable potential as a bioenergy crop, owing to low inputs, high yields, soil carbon sequestration and possible localised ecosystem services benefits. A small positive net margin for the *Miscanthus* heat chain is driven by reduced heating costs compared with oil, but belies the poor financial performance of *Miscanthus* as a crop for farmers. Low farm gate prices for *Miscanthus* biomass, high establishment costs and the risk premium associated with 20-year plantation lifetimes, act as major barriers for farm uptake (Zimmerman et al., 2013). Another bottleneck is the cost of small-scale pellet processing in the absence of an established market. Farmers receive €75 Mg^{-1} DM at the farm gate, compared with a delivered pellet price of €329 Mg^{-1} DM, reflecting high processing costs but also an opportunity to generate economic activity within rural regions. Further incentivisation of this crop at the farm level would represent better value for money than indiscriminant encouragement of less sustainable bioenergy options via FiTs and mandatory biofuel blend targets.

We conclude that consequential life cycle assessment and ecosystem services screening should be integrated into sustainability assessment criteria for renewable energy subsidies, so that public money is directed towards more sustainable options that support resource efficiency, climate protection and ecosystem services provisioning.

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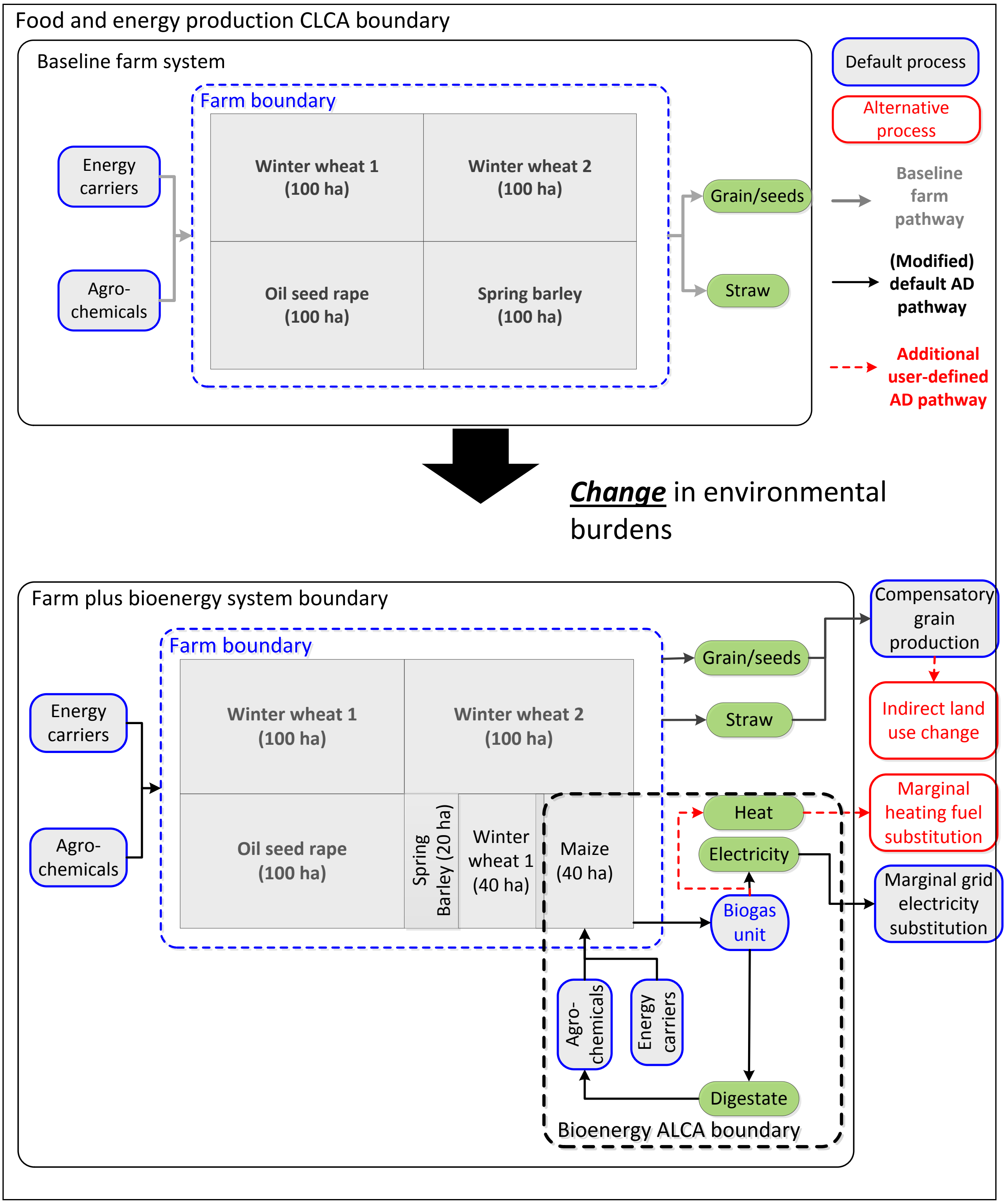
Figure titles

Figure 1. Main material flows and processes occurring in the baseline arable farm (above), and in the maize-in-rotation AD scenario (AD-MZrot), following rotation optimisation (below), with attributional and consequential LCA boundaries shown. Nutrient cycling and emissions associated with the recycling of digestate are captured within the arable farm system.

Figure 2. Environmental burden changes expressed as a percentage of baseline arable farm burdens under default settings (including 50% iLUC) for each AD scenario described in Table 4, plus a variation of the default A-F scenario with landfilling instead of composting as the counterfactual waste management option. Lower bars represent best case AD design and management plus use of all CHP-heat while upper bars represent worst case AD design and management.

Figure 3. Main factors contributing to GWP (a), EP (b), AP (c) and FRDP (d) burden changes relative to baseline farm system across scenarios, including avoided (A) and displaced (D) processes, expressed per Mg dry matter of bioenergy feedstock (scenarios from which values derived in brackets). Net burden changes per Mg DM are reported for each feedstock above the x axis.

Figure 4. Net GWP change per hectare of bioenergy crop cultivation across the different scenarios, after attributing 0%, 50% and 100% iLUC to displaced food production, based on iLUC Method 1 (default) and alternative iLUC method 2 (see S3.3). Negative values represent GHG abatement. Error bars represent worst-to-best case AD design and management.



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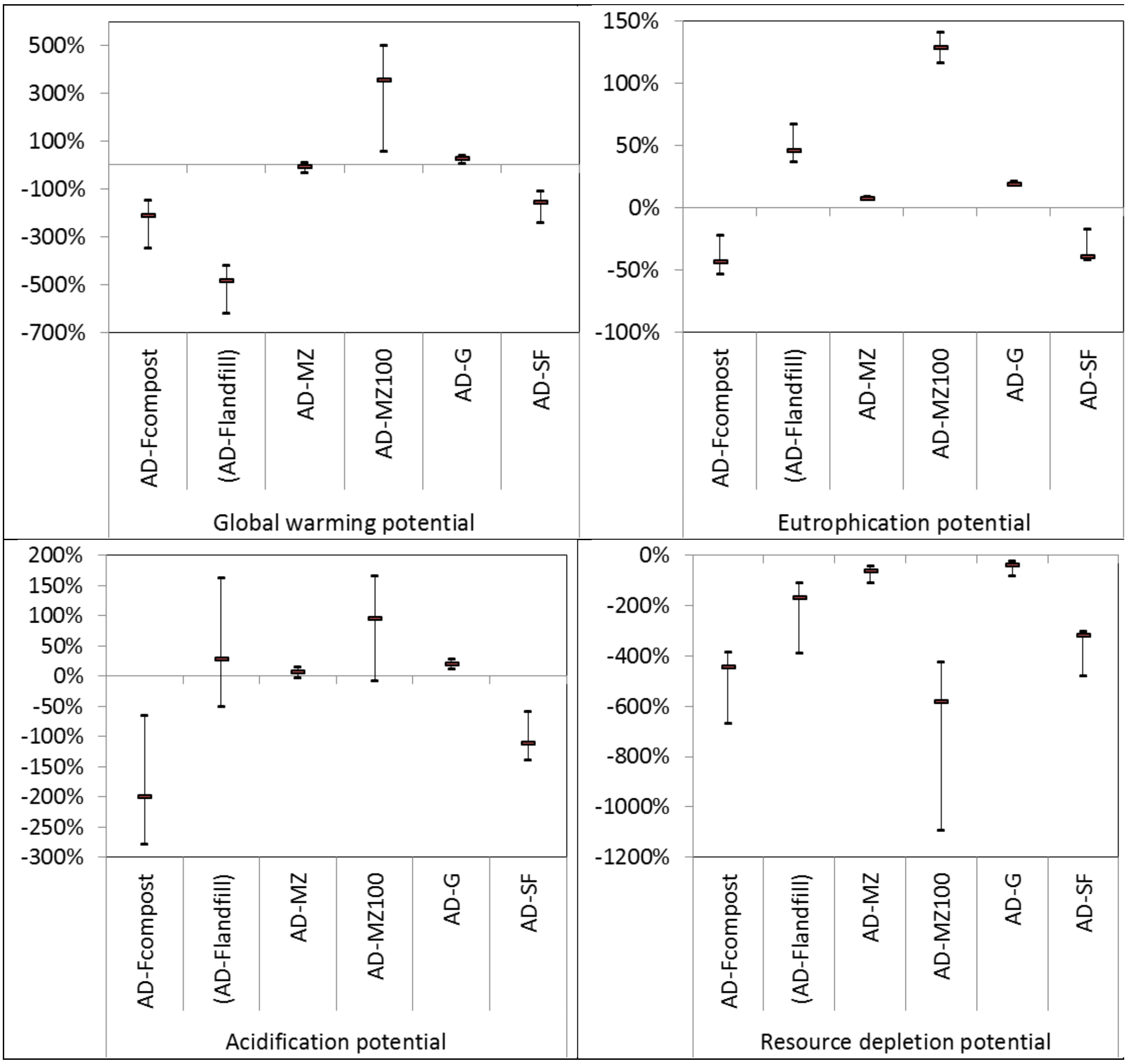

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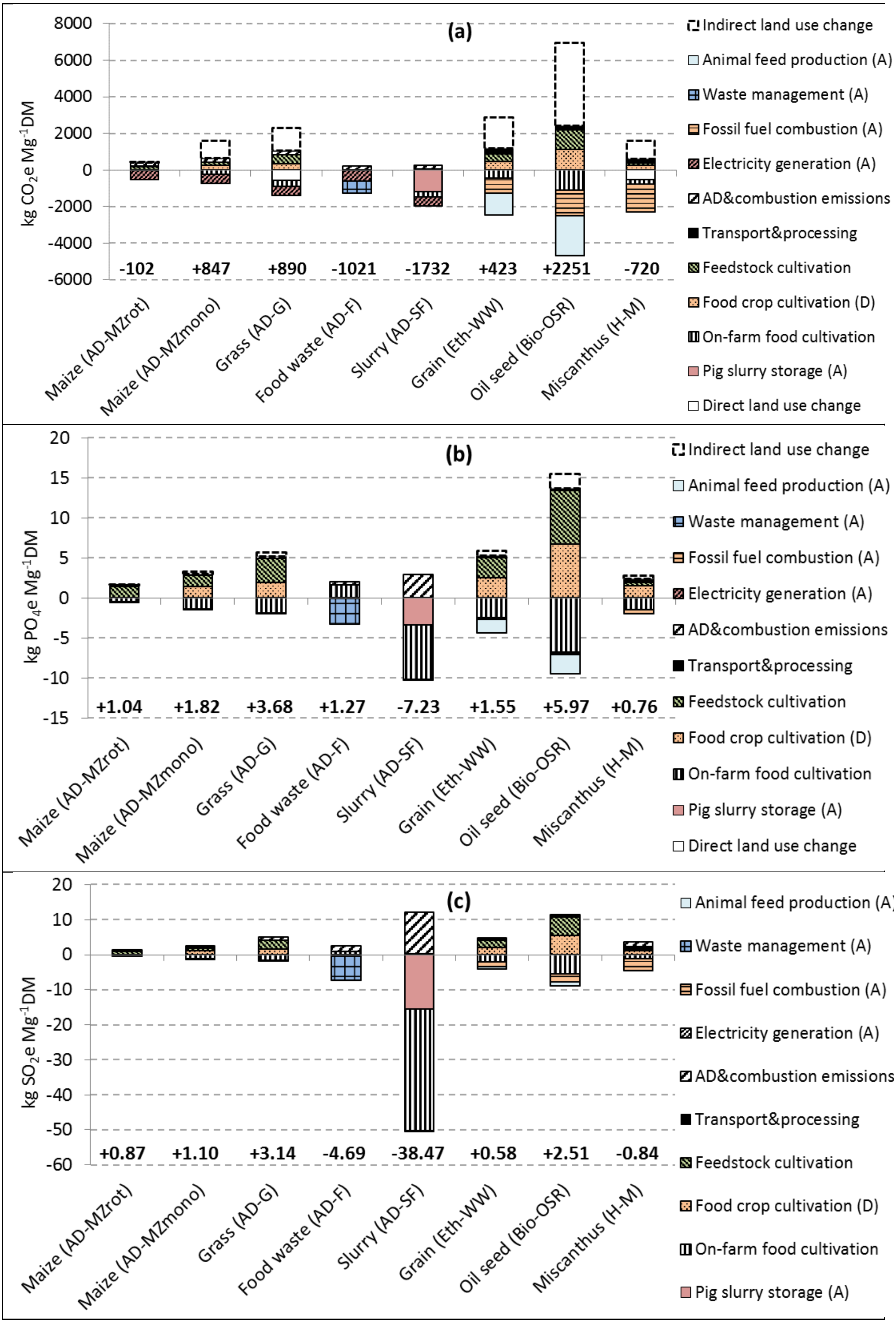
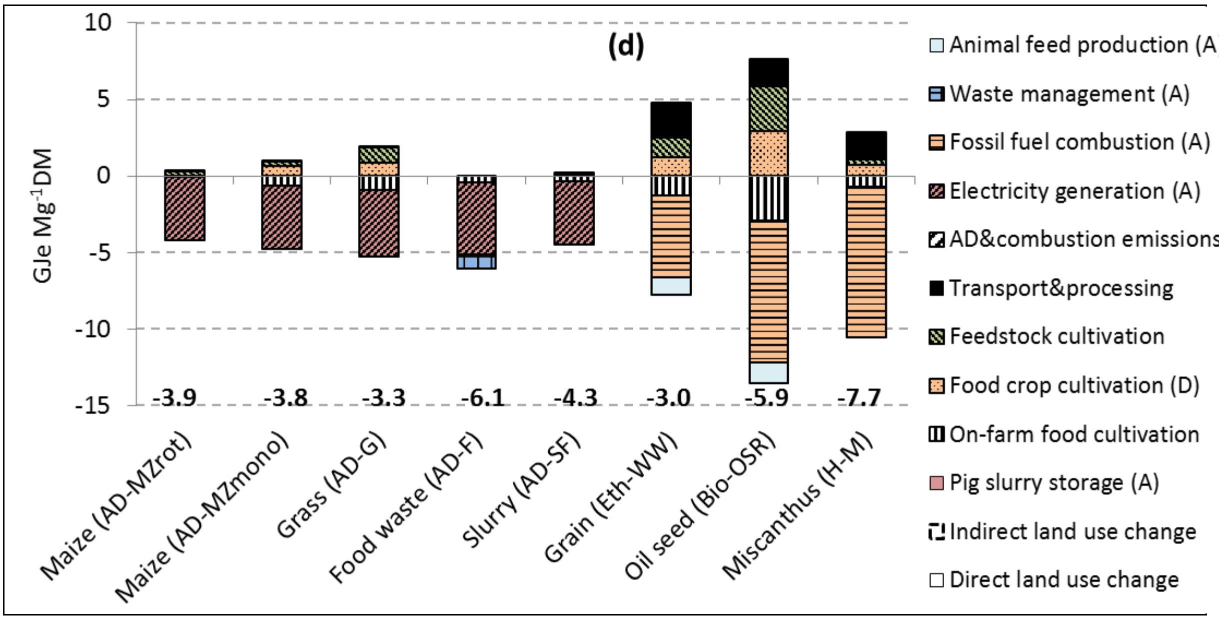


(a)
kg CO2e Mg-1DM
8000
6000
4000
2000
0
-2000
-4000
-6000
-102
+847
+890
-1021
-1732
+423
+2251
-720
Maize (AD-MZrot)
Maize (AD-MZmono)
Grass (AD-G)
Food waste (AD-F)
Slurry (AD-SF)
Grain (Eth-WW)
Oil seed (Bio-OSR)
Miscanthus (H-M)
Indirect land use change
Animal feed production (A)
Waste management (A)
Fossil fuel combustion (A)
Electricity generation (A)
AD&combustion emissions
Transport&processing
Feedstock cultivation
Food crop cultivation (D)
On-farm food cultivation
Pig slurry storage (A)
Direct land use change
(b)
kg PO4e Mg-1DM
20
15
10
5
0
-5
-10
-15
+1.04
+1.82
+3.68
+1.27
-7.23
+1.55
+5.97
+0.76
Maize (AD-MZrot)
Maize (AD-MZmono)
Grass (AD-G)
Food waste (AD-F)
Slurry (AD-SF)
Grain (Eth-WW)
Oil seed (Bio-OSR)
Miscanthus (H-M)
Indirect land use change
Animal feed production (A)
Waste management (A)
Fossil fuel combustion (A)
Electricity generation (A)
AD&combustion emissions
Transport&processing
Feedstock cultivation
Food crop cultivation (D)
On-farm food cultivation
Pig slurry storage (A)
Direct land use change
(c)
kg SO2e Mg-1DM
20
10
0
-10
-20
-30
-40
-50
-60
+0.87
+1.10
+3.14
-4.69
-38.47
+0.58
+2.51
-0.84
Maize (AD-MZrot)
Maize (AD-MZmono)
Grass (AD-G)
Food waste (AD-F)
Slurry (AD-SF)
Grain (Eth-WW)
Oil seed (Bio-OSR)
Miscanthus (H-M)
Animal feed production (A)
Waste management (A)
Fossil fuel combustion (A)
Electricity generation (A)
AD&combustion emissions
Transport&processing
Feedstock cultivation
Food crop cultivation (D)
On-farm food cultivation
Pig slurry storage (A)



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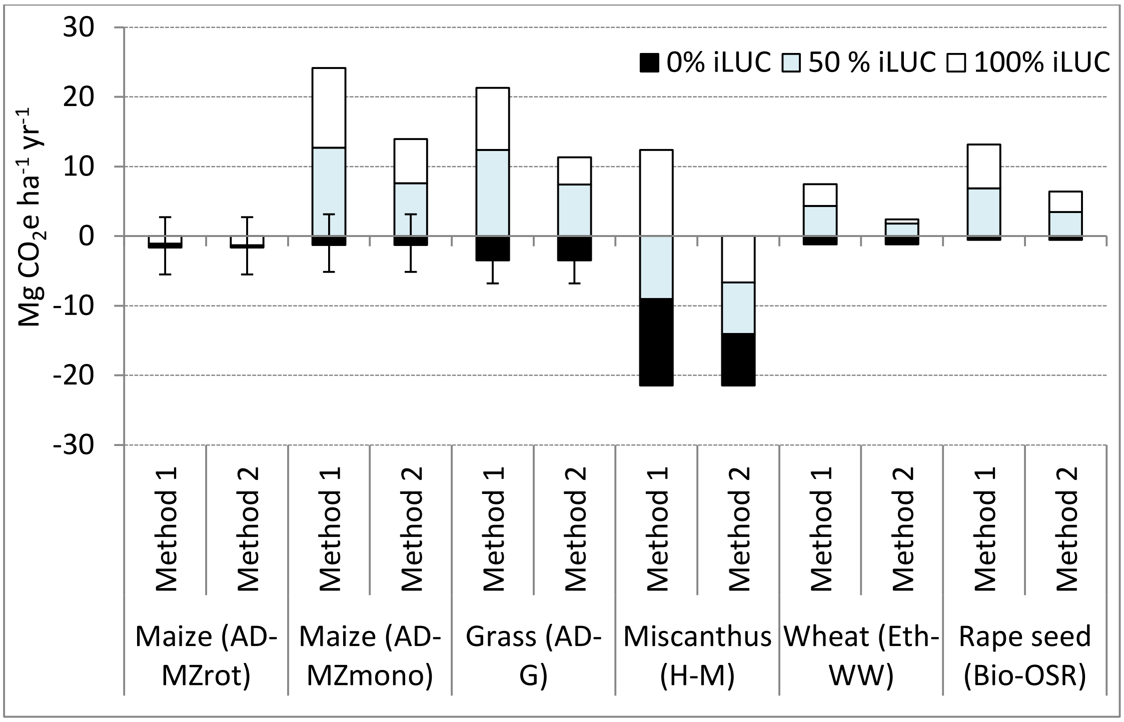

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Table 1. Environmental burdens attributed to upstream and counterfactual processes

Input	Reference unit	Global warming potential kg CO_2e	Eutrophication potential kg PO_4e	Acidification potential kg SO_2e	Resource depletion potential MJe
<u>Fertilizers and other agrochemicals</u>					
Ammonium nitrate-N	kg N	6.10	0.0068	0.024	55.7
Triple superphosphate	kg P_2O_5	2.02	0.045	0.037	28.3
Potassium chloride K_2O	kg K_2O	0.50	0.0008	0.0017	8.32
Lime	kg $CaCO_3$	2.04	0.0004	0.0007	3.31
Crop protection products	kg active ingredient	10.1	0.033	0.097	174
<u>Sources of fuel/energy</u>					
Marginal electricity generated	kWh_e	0.42	0.00006	0.00023	7.32
Oil heating	kWh_{th}	0.34	0.00011	0.00075	4.55
Diesel	MJ LHV	0.087	0.00002	0.00014	1.20
Petrol	MJ LHV	0.090	0.00023	0.00016	1.22
Transport	tkm	0.081	0.00007	0.00030	1.06
<u>Avoided animal feed</u>					
Soybean meal*	kg DM	0.094	0.0039	0.0018	6.82
Maize silage	kg DM	0.168	0.0015	0.0037	0.329
Palm oil	Kg oil	2.33	0.0057	0.0084	0.006
<u>Avoided food waste management</u>					
Landfilling	kg waste	517	0.14	0.42	-1563
Composting	kg waste	170	0.83	1.81	500

* Accounts for substitution of palm oil with soy-oil. Data based on Ecoinvent (2010), DEFRA (2012), CFT (2012), and Styles et al. (2014) for avoided waste management.

Table 2. Default "D" (in bold), best- "B" and worst- "W" case parameters applied to generate the main results in this study.

Baseline farm slurry application*	AD design and management (Table 6)	Excess** AD heat output utilised	Digestate application method	Displaced food and animal feed production incurring iLUC	Food waste counterfactual management
Splash plate[D]	Best case[B]	**0%**[W,D]	**Trailing shoe**[B]	0% [B]	**Composting**[W,D]
Trailing shoe	Good default	50%	Splash plate[W]		Landfilling[B]
	Default[D]	100%[B]		**50%**[D]	
	Poor default			100% [W]	
	Worst case[W]				

*Pig slurry arable farm baseline only (BL-AP)
**Remaining available AD heat output after farm and farmhouse heating supplied
Default permutations in bold

Table 3. Direct emission factors applied in the farm model, across baseline farms and bioenergy scenarios

Process	Unit	CO_2	CH_4	N_2O-N	NH_3-N	NO_x	NO_3-N	P
Fertiliser-N application	Fraction N			[1]0.01	[2]0.018		[3]0.1	
Crop residue N application	Fraction TN			[1]0.01			[3]0.1	
Manure-/digestate- application	Fraction TN			[1]0.01	[4]0.08 – 0.27		[4]0 – 0.28	
All P amendments	Fraction P							[6]0.01
Lime application	kg per kg lime	[1]0.44						
Tractor diesel combustion	kg per kg diesel	[7]3.05	[7]0.000044	[7]0.000048		[8]0.004		

[1]IPCC (2006); [2]Misselbrook et al. (2012); [3]Duffy et al. (2013); [4]MANNER-NPK outputs (Nicolson et al., 2013); [5]Webb and Misselbrook (2004); [6]Withers, pers. comm. (2013); [7]DEFRA (2012); [8] Dieselnet (2013).

Table 4. Key features of the eight tested bioenergy scenarios

Scenario name	Feedstock	CHP capacity	Bioenergy area	Slurry (4% DM)	Maize (30% DM)	Grass (25% DM)	Food waste (26% DM)	Miscanthus (DM basis)	Winter wheat grain (85% DM)	Rape seed (85% DM)	Direct land use change
		kWe	**ha**	**Mg yr^{-1} to bioenergy**							
AD-F	**Food waste**	561	0				10 000				
AD-MZ	**Maize in rotation**	1000*	40		1800						
AD-MZ100	**Maize monoculture**	929	400		18 000						
AD-G	**Grass**	1000**	40			1600					40 ha arable to grass
AD-SF	**Pig slurry, food waste**	343	0	5098			6000				
H-M	**Miscanthus**	NA	40					504			40 ha arable to miscanthus
Eth-WW	**Winter wheat**	NA	100						875		
Bio-OSR	**Oil seed rape**	NA	100							330	

BL = baseline farm scenario (400 ha arable farm)
BE = bioenergy
*Central AD unit supplied by 19 370 t maize annually, produced on 40 ha in each of 10.8 supply farms modelled on the baseline arable farm
** Central AD unit supplied by 23 302 t grass annually, produced on 40 ha in each of 14.6 supply farms modelled on the baseline arable farm

Table 5. Burden changes relative to the baseline farm system, expressed in kg or GJ equivalents and as a percentage, excluding land use change, and also as a percentage including 50% land use change where relevant

	AD-F	AD-MZ_{rot}	A-MZ_{mono}	AD-G	AD-SF	H-M	Eth-WW	Bio-OSR
kg CO_2e	-2,654,793	-66,354	-504,701	-139,264	-858,847	-118,441	-54,189	-1,946,164
	-209%	-5%	-40%	-11%	-67%	-9%	-4%	-152%
(50% iLUC)	-209%	-4%	+359%	+28%	-28%	+25%	+50%	-152%
kg PO_4e	-3,295	+559	+7,832	+1,281	+189	+1,191	+1,363	-3,452
	-43%	+7%	+103%	+17%	+2%	+16%	+18%	-39%
(50% LUC)	-43%	+7%	+129%	+19%	+5%	+15%	+22%	-39%
kg SO_2e	-12,202	+470	+5,937	+1,256	-424	+199	+705	-15,167
	-199%	+8%	+97%	+21%	-7%	+3%	+12%	-248%
GJe	-32,940	-4,376	-43,218	-2,781	-7,950	-3,875	-3,456	-21,589
	-442%	-59%	-581%	-37%	-107%	-52%	-46%	-290%

Table 6. Theoretical CO_2e abatement costs required for non-subsidised supply chains to break even, before and after attributing iLUC to 50% of displaced food production, where negative values represent potentially profitable bioenergy value chains before subsidies, and NA represents no GHG abatement for the scenario. Also shown is life cycle GWP per MJ biofuel (biogas, transport biofuel and heating pellets) produced in each scenario, calculated according to ALCA and CLCA methods, and default Renewable Energy Directive ALCA GWP values (bottom row).

	Method	**iLUC**	**Use all AD heat**	**AD-F**	**AD-MZ_{rot}**	**AD-MZ_{mono}**	**AD-G**	**H-M**	**AD-SF**	**Eth-WW**	**Bio-OSR**
€ Mg^{-1} CO_2e avoided	CLCA	None	No	-5	775	1189	459	-38	9	739	578
	CLCA	50%	No	-5	930	NA	NA	-90	9	NA	NA
	CLCA	None	Yes	-70	-23	11	65	-38	-56	739	578
	CLCA	50%	Yes	-70	-24	NA	NA	-90	-56	NA	NA
g CO_2e MJ^{-1} biofuel produced	CLCA	None	NA	-35	31	34	14	-10	-42	73	75
	CLCA	50%	NA	-35	33	112	113	45	-42	136	226
	ALCA	None	NA	-18	34	34	14	-10	18	35	61
	ALCA-RED default values (EC,2009)	None		3					4	52	56

Table 7. Ecosystem services effects for each of the scenarios involving bioenergy crop cultivation. In this traffic light assessment, green and red represent delivery of services and disservices, respectively. Orange represents either mixed service and disservice delivery from the respective land use, or inconclusive outcomes dependent on specific farm management decisions. Plus and minus characters depict the expected direction and value of an impact (Table S6.2).

Ecosystem services		AD-MZ_{rot}	AD-MZ_{mono}	AD-G	H-M	Eth-WW	Bio-OSR
		Maize	**Maize**	**Grass**	**Misc**	**Wheat**	**OSR**
		40 ha	400 ha	40 ha	40 ha	100 ha	100 ha
Provisioning services	1.1 Food	+/-	---	-	-	--	--
	1.2 Fodder	---	---	---	---	+/-	+/-
	1.3 Biomass for energy	+++	+++	++	+++	+	+
	1.4 Water supply	+/-	+/-	+/-	-	+/-	+/-
	1.5 Wild food and genetic resources	+/-	+/-	+/-	+/-	+/-	+/-
	1.6 Carbon	--	--	+/-	++	--	--
Regulation services	2.1 Hazard regulation	---	---	+/-	++	---	---
	2.2 Regulation of water quantity	--	--	+	++	+/-	+/-
	2.3 Climate regulation	+	+/-	+/-	++	+/-	+/-
	2.4 Waste breakdown	+/-	+/-	+/-	-	+/-	+/-
	2.5 Purification in soil	--	--	-	+	--	--
	2.6 Disease and pest regulation	-	-	-	+/-	-	-
	2.7 Pollination	-	-	-	+/-	-	+/-
Cultural services	3.1 Environmental settings – socially valued landscapes	+/-	--	+	+/-	+/-	+/-
	3.2 Wild species diversity and wildlife habitat	-	-	-	+/-	-	-